

# Carbon dioxide evolution in runoff from simulated rainfall on long-term no-till and plowed soils in southwestern Ohio

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## Abstract

Water erosion results in the mobilization and depletion of soil organic carbon (SOC), but studies providing direct experimental evidence of eroded C mineralization and its linkage to the global C cycle are lacking. A study was conducted to determine the mineralization of SOC in runoff from a southwestern Ohio Crosby soil (fine, mixed, mesic Aeric Ochraqualf) that had been under no-till (NT), chisel till (CT) and moldboard plow (MP) for 38 years. To simulate present and future soil erosion conditions, the 0–3 and 5–8 cm soil layers from triplicate soil blocks extracted from each tillage practice were used. Soil layers were transferred to runoff trays and simulated rainfall ( $30 \pm 5 \text{ mm h}^{-1}$ ) was applied for 1 h. Runoff was collected at 20, 40 and 60 min following initiation of rainfall event. Carbon dioxide production was monitored for 100 days in runoff samples incubated without and with N ( $0.15\text{--}0.20 \text{ g NH}_4\text{-N kg}^{-1}$  sediment) amendment to simulate situations where water erosion coincides with high soil mineral N availability. The data show that soil layer depth and time of runoff sampling had no significant effect on any of the parameters considered. Sediment delivery ( $\text{g sediment m}^{-2} \text{ h}^{-1}$ ) was significantly lower in the NT (17.4) than in the other tillage (CT: 48.9, MP: 34.1) practices; it was inversely related to degree of soil aggregation and sediment C content. The average of C export ranged between 0.6 and  $1.1 \text{ g C m}^{-2} \text{ h}^{-1}$ , and was highest in the CT soil. Mineralization of runoff C followed a first-order kinetics and proceeded at rates significantly higher in NT and N-amended runoff. Nearly half of the total runoff C mineralization recorded during the entire incubation occurred during the first 20 days suggesting that under field conditions, most of the erosion-induced  $\text{CO}_2$  production will occur in the days immediately following rainfall events. Potentially mineralizable carbon ( $\text{C}_0$ ) in runoff was significantly greater with NT ( $13.5 \text{ g C kg}^{-1}$  sediment) than with the other tillage practices, and was positively correlated with sediment C content ( $R^2 = 0.76$ ) and cumulative C mineralized ( $R^2 = 0.77\text{--}0.97$ ). The data presented showed that 29–35 and 33–46% of the C exported in runoff was mineralized in 100 days without and with N amendment, respectively. These results are the first quantitative evidence that a significant fraction of the SOC released through water erosion is mineralizable and, therefore represents an additional source of atmospheric  $\text{CO}_2$ .  
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## 1. Introduction

Water erosion controls soil organic carbon (SOC) reserves and distribution across terrestrial landscapes, and could therefore have major implications for terrestrial C cycling, atmospheric carbon dioxide ( $\text{CO}_2$ ),

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and climate warming. Through its association with soil minerals and occlusion within aggregates (Beare et al., 1994), SOC is physically protected against biological mineralization. Soil aggregates are of various sizes with the larger generally less resistant to external disruption (Rasiah et al., 1992), and containing more labile C than the microaggregates (Jastrow et al., 1996). The physical protection of SOC afforded by aggregation is removed by water erosion when rain-impacted aggregates are broken down and labile SOC fractions are released. Furthermore, the selective removal of the most labile SOC fractions by water erosion strongly suggests that CO<sub>2</sub> emission from soils into the atmosphere could be enhanced during and immediately after erosional events, but so far this has not been investigated.

There is an abundant literature relating soil erosion to ecosystem productivity decline (see review by Olson et al. (1994)), impairment of development infrastructures (Evans et al., 2000) and water quality degradation resulting from delivery of non-point source pollutants to water bodies (Logan, 1993). However, data regarding the fate of SOC mobilized by water erosion are limited, and are primarily derived from review of published SOC data (Lal, 1995; Jacinthe et al., 2001a), modeling exercises (Lee et al., 1996; Jacinthe and Lal, 2001), and inference from the spatial distribution of chemically defined classes of organic compounds across eroding landscapes (Beyer et al., 1993). The lack of experimental data has also lead to conflicting interpretation of the information available. While Lal (1995) and Jacinthe et al. (2001a) suggest that the mineralization of aggregate-protected SOC released during erosional events could contribute to additional CO<sub>2</sub> input to the atmosphere, others (Van Noordwijk et al., 1997; Stallard, 1998) proposed that erosion could lead to C sequestration since most eroded SOC is retained in depositional areas where biological decomposition is constrained by a lack of oxygen. A recent study (Jacinthe et al., 2001b) has provided evidence of labile C enrichment in eroded materials retained in terrestrial deposits downslope of cultivated landscapes. Indication of labile C accumulation in depositional sites has also been reported by Gregorich (1996). These studies, however, provided no estimates of historical mineralization of eroded C during sediment transport and deposition. Therefore, the fate of eroded SOC

(mineralization versus sequestration), the net effect of erosion on CO<sub>2</sub> production and the global C cycle remain uncertain, and these uncertainties are acknowledged in recent IPCC (1996, 2000) reports.

The transport, and the physico-chemical attributes of SOC mobilized by water erosion have been examined in several studies. These include investigation of (i) the effect of various soil covers (Lowrance and Williams, 1988) and tillage practices (Schreiber and McGregor, 1979) on SOC exports and the relative proportion of particulate organic carbon (POC) in the C exported, (ii) the export of dissolved organic carbon (DOC) from cultivated landscapes in relation to soil erosion history (Bajracharya et al., 1998), and (iii) the distribution of eroded C among various sediment particle sizes as a function of rainfall energy (Palis et al., 1997). With the exception of an evaluation of biological oxygen demand (BOD) in runoff samples by Schreiber and McGregor (1979), these studies did not examine the biological availability and fate of eroded C. Therefore, direct quantitative data regarding the biological quality of eroded SOC and its mineralization during translocation remain unavailable. Yet, this information is the first step towards improving our understanding of the processes affecting the dynamics of eroded C, and reducing the uncertainties concerning water erosion impacts on the global C cycle.

Research has shown that cultivation results in loss of soil aggregation (Hussain et al., 1999) and depletion of labile SOC pools (Alvarez et al., 1995). Conversely, management practices such as no-tillage contribute to the restoration of soil physical and biological properties (Alvarez et al., 1995; Hussain et al., 1999), and increased pools of aggregate-protected C (Beare et al., 1994; Wander and Bidart, 2000). In addition, labile C fractions were found to accumulate preferentially at the soil surface in no-till and reduced tillage practices, whereas distribution was more uniform in plowed soils as a result of frequent mixing of crop residue with the mineral soil layers (Alvarez et al., 1995).

From the foregoing, it is hypothesized that the fate of erosion-displaced SOC depends on its biochemical quality, an attribute strongly linked to C quality in the terrestrial ecosystem from which eroded C originates. In other words, a larger fraction of C in runoff originating from no-tillage plots is susceptible to mineralization than in runoff from intensively tilled soils. A second hypothesis of the study is that the

dynamics of runoff C is dependent on the depth of soil being eroded, with no-tillage exhibiting a stronger depth variability than the other tillage systems. It is further hypothesized that, through its stimulating effect on microbial activity, mineral N availability enhances runoff C mineralization and CO<sub>2</sub> production.

The present study was undertaken with the following two objectives: (i) estimating erosional SOC exports under three different long-term tillage practices: no-till (NT), chisel till (CT) and moldboard plow (MP), and (ii) determine CO<sub>2</sub> evolution in runoff in relation to soil management, depth of soil eroded and mineral N availability.

## 2. Materials and methods

### 2.1. Collection of soil blocks

Soil blocks were collected in September 2000 at the western branch of the Ohio Agricultural Research and Development Center (OARDC) in south Charleston, OH (39°45'N, 83°36'W) from the long-term (established in 1962) tillage plots, including NT, CT and MP. Soils, developed from medium-textured Wisconsinan glacial till, are predominantly silt-loam of the Crosby (fine, mixed, mesic Aeric Ochraqualf) and Brookston series (fine-loamy, mixed, mesic, typic Argiaquolls). In the NT plots, corn is sown directly into the previous crop residue with a NT planter. The CT plots are chisel-tilled to a depth of 20–25 cm prior to planting of

corn. The MP treatment consists of soil plowing and inversion to a depth of 30 cm in the fall, followed by disking and harrowing prior to corn seeding in the spring. Further details regarding the long-term tillage experiment are provided in previous reports (Dick et al., 1991; Mahboubi et al., 1993).

Three soil blocks were taken from each tillage treatment using frames made of aluminum metal sheet. Soil blocks (45 cm long × 30 cm wide × 10 cm deep) were transported to the laboratory, and moistened daily to prevent excessive drying and development of cracks. Soil blocks were used within 15 days of their collection in the rainfall simulation experiment. Gravimetric moisture content of soils at the time of sampling averaged  $0.19 \pm 0.03$  kg H<sub>2</sub>O kg<sup>-1</sup> soil. Composite soil samples were also collected for determination of physico-chemical and biological properties of soils (Table 1).

### 2.2. Runoff experiment

Residue was removed from the surface of soil blocks. Each soil block was cut into two layers (0–3 and 5–8 cm) that were then transferred into a runoff tray (40 cm × 30 cm). The bottom of the runoff tray was fitted with cheese cloth and a sheet of plexiglass at the center of which was a drainage hole. The soil layer in the tray was moistened overnight from the bottom through a tygon tubing connected to the drainage hole. The soil in the tray was allowed to freely drain 1–2 h prior to commencing the rainfall experiment.

Table 1  
Selected physico-chemical (0–10 cm) and biological characteristics of the soils used

Parameter	Tillage practices <sup>a</sup>		
	NT	CT	MP
pH <sub>(water)</sub>	4.7 ± 0.01 <sup>b</sup>	5.1 ± 0.01	6.8 ± 0.03
Total aggregation (%)	62.1 ± 5.3	20.5 ± 0.5	18.9 ± 1.9
Mean weight diameter (mm)	1.8 ± 0.20	0.3 ± 0.02	0.2 ± 0.01
Total nitrogen (g N kg <sup>-1</sup> soil)	2.9 ± 0.07	1.2 ± 0.00	1.0 ± 0.00
Total organic carbon (g C kg <sup>-1</sup> soil)	30.0 ± 0.51	11.2 ± 0.00	9.4 ± 0.07
Basal soil respiration (mg CO <sub>2</sub> -C kg <sup>-1</sup> soil per day) <sup>c</sup>	4.5 ± 1.9	1.8 ± 0.7	0.7 ± 0.5
Microbial biomass carbon (mg C kg <sup>-1</sup> soil) <sup>d</sup>	262.8 ± 8.5	98.8 ± 6.9	36.1 ± 5.6

<sup>a</sup> NT: no-till; CT: chisel till; MP: moldboard plow.

<sup>b</sup> Values are mean ± S.D.

<sup>c</sup> Rate of CO<sub>2</sub> evolution in field-moist soil samples (<2 mm) during the first 10 days of incubation (22 °C).

<sup>d</sup> Determined by the chloroform fumigation method using an unfumigated control and a microbial biomass mineralization coefficient  $k_C = 0.45$ .

To generate runoff, rainfall simulation using the equipment described by Choudhary et al. (1997) was employed. Tap water was supplied at a pressure 50 kPa to the rainfall simulator nozzles (V-jet 80/100). The nozzles oscillation frequency was set at 60 sweeps  $\text{min}^{-1}$  to achieve a rainfall intensity of  $30 \pm 5 \text{ mm h}^{-1}$ . Rainfall intensity–duration–frequency analysis for Ohio (Schwab et al., 1993) indicates that this intensity is typical 1 h storms with a return period of 1–2 years. Analysis of long-term rainfall data from the North Appalachian Experimental Watersheds in Coshocton, OH (Bonta and Rao, 1994) has also shown that 1 h storms occurring during the growing season (April–September) with a return frequency of 5 years or less have intensities between 25 and 40  $\text{mm h}^{-1}$ . Therefore, the rainfall intensity selected for this simulation study is representative of frequent erosive rainfalls affecting Ohio soils.

The runoff trays were positioned 210 cm below the nozzles on a metallic frame with adjustable slope. There were two runoff trays per nozzle. Runoff was channeled through the tray outlet into a 1.2 l acid-washed collection jar. Each rainfall event lasted 1 h, and runoff from each tray was collected at 20, 40 and 60 min. Collected runoff was thoroughly mixed and, from each collection jar, two runoff subsamples (40 ml each) were taken for the incubation study. The remaining runoff was dried in a forced-air oven (105 °C), and the volume and sediment content of runoff computed. Dried sediment was ground, sieved (250  $\mu\text{m}$ ) and analyzed for total organic C content.

### 2.3. Incubation of runoff samples

The 40 ml runoff subsamples taken from the collection jar were transferred into 120 ml incubation bottles. Half of the incubation bottles were amended with 5 ml of a  $\text{NH}_4\text{-N}$  solution (10  $\text{mg N l}^{-1}$ ), while the other half was not. Incubation bottles were stoppered with butyl rubber septa and crimp-sealed. Evolution of  $\text{CO}_2$  in the incubation bottle headspace was monitored over a period of 100 days. Total  $\text{CO}_2$  production was computed as the sum of  $\text{CO}_2$  in the gas and liquid phases using the procedure described in Jacinthe and Groffman (2001). The  $\text{CO}_2$  production data were fitted to a first-order kinetic model:

$$C_t = C_0(1 - e^{-kt}) \quad (1)$$

where  $C_t$  is the cumulative  $\text{CO}_2$  ( $\text{g CO}_2\text{-C kg}^{-1}$  sediment) evolved at time  $t$  (days),  $C_0$  the initial mineralizable C ( $\text{g C kg}^{-1}$  sediment) in runoff and  $k$  is the rate constant (per day).

### 2.4. Analytical

Finely ground (250  $\mu\text{m}$ ) soil and sediment samples were analyzed for organic C content by combustion at 900 °C with a C–N analyzer (Carlo-Erba NC 2100). The analysis of  $\text{CO}_2$  was performed using a Shimadzu GC-14A gas chromatograph equipped with a thermal conductivity detector and a 90  $\text{cm} \times 0.3 \text{ cm}$  Hayesep D column (Alltech, IL). The carrier gas was ultrahigh purity helium (20  $\text{ml min}^{-1}$ ). The column and detector temperatures were set 50 and 150 °C, respectively.

### 2.5. Statistical

The data were statistically analyzed using ANOVA for a completely randomized design with soil tillage, soil depth, nitrogen addition, and runoff collection time as class variables. Response variables included sediment production and delivery rate, C content of sediment and C export rate, cumulative and rate of runoff C mineralization. Statistical significance was determined at the 95% confidence level. Data analysis and model fitting were performed using the procedures GLM and NLIN, respectively, available in SAS (SAS Institute, 1990).

## 3. Results

### 3.1. Sediment production and carbon content

Statistical analyses revealed significant effects of soil tillage practices on sediment delivery rate ( $P < 0.001$ ), and sediment C content ( $P < 0.01$ ). However, soil depth had no significant effect on these parameters. Sediment delivery rate did not vary significantly during the 1 h rainfall event; however, sediment C content showed a slight increase during the rainfall event (Table 2).

Sediment C enrichment ratios varied between 0.9 and 3, and averaged 1.4, 2.3 and 1.7 in the NT, CT and MP soils, respectively. Sediment from the NT soil contain twice as much C (27.2–61.9  $\text{g C kg}^{-1}$  sediment) than

Table 2  
Sediment carbon content and delivery rate as affected by tillage practices

Parameter	Sampling time (min)	Tillage practices <sup>a</sup>			LSD <sup>b</sup>
		NT	CT	MP	
Sediment C content (g C kg <sup>-1</sup> sediment)	0–20	36.2 ± 6.9 <sup>c</sup>	20.9 ± 3.9	18.3 ± 1.2	3.9
	20–40	42.7 ± 7.8	22.9 ± 3.4	18.6 ± 0.9	
	40–60	47.8 ± 10.9	22.1 ± 1.9	18.9 ± 1.4	
	Tillage mean	42.3 ± 9.9	22.0 ± 3.3	18.6 ± 1.2	
C enrichment ratio	0–20	1.2 ± 0.2	2.2 ± 0.4	1.6 ± 0.1	0.2
	20–40	1.4 ± 0.3	2.4 ± 0.4	1.7 ± 0.1	
	40–60	1.6 ± 0.4	2.3 ± 0.2	1.7 ± 0.1	
	Tillage mean	1.4 ± 0.3	2.3 ± 0.3	1.7 ± 0.1	
Sediment delivery rate (g sediment m <sup>-2</sup> h <sup>-1</sup> ) <sup>d</sup>	0–20	18.9 ± 8.2	42.5 ± 12.0	39.0 ± 16.6	11.2
	20–40	11.7 ± 4.2	45.8 ± 23.4	29.9 ± 19.2	
	40–60	21.7 ± 8.8	35.3 ± 15.9	33.4 ± 18.6	
	Tillage mean	17.4 ± 8.5	48.9 ± 25.0	34.1 ± 18.6	
Rate of C transport (g C m <sup>-2</sup> h <sup>-1</sup> ) <sup>d</sup>	0–20	0.7 ± 0.3	1.4 ± 0.9	0.7 ± 0.3	0.3
	20–40	0.5 ± 0.2	1.1 ± 0.5	0.6 ± 0.4	
	40–60	1.0 ± 0.4	0.8 ± 0.4	0.6 ± 0.3	
	Tillage mean	0.7 ± 0.4	1.1 ± 0.7	0.6 ± 0.3	

<sup>a</sup> NT: no-till; CT: chisel till; MP: moldboard plow.

<sup>b</sup> LSD for comparison of the mean values among tillage practices.

<sup>c</sup> Values are mean ± S.D.

<sup>d</sup> Divide by 30 mm h<sup>-1</sup> to convert to g m<sup>-2</sup> mm<sup>-1</sup> of rainfall.

sediment from the CT (15.5–28.5 g C kg<sup>-1</sup> solid) and MP (16.5–20.7 g C kg<sup>-1</sup> sediment) soils. Runoff soil loss was highest in the MP soils with an average sediment delivery rate of 48.9 g sediment m<sup>-2</sup> h<sup>-1</sup> which was 1.4–2.8-fold that from the CT and NT soils (Table 2). Sediment delivery rate was inversely related with sediment C content ( $R^2 = 0.58$ ) and with the aggregation indices, percentage of aggregation ( $R^2 = 0.68$ ) and mean weight diameter of soil aggregates ( $R^2 = 0.63$ , Tables 1 and 2).

During the simulated rainfall event, C exports averaged 0.7, 1.1 and 0.6 g C m<sup>-2</sup> h<sup>-1</sup> (0.02, 0.04, 0.02 g C m<sup>-2</sup> mm<sup>-1</sup> rainfall) in the NT, CT and MP soils, respectively (Table 2). The higher C export in the CT than in the other soils resulted from the combination of high sediment load and moderate sediment C content.

### 3.2. Mineralization of eroded organic carbon

The data on CO<sub>2</sub> evolution (Fig. 1) in runoff samples fitted very well to the first-order model as evidenced by values for  $R^2 > 0.97$  (Table 3). Convergence occurred after less than 12 iterations in all cases.

Estimates of  $C_0$  (potentially mineralizable C) and  $k$  (rate constant) obtained from data fitting were submitted to ANOVA. The analysis showed that soil depth and time of runoff sampling had no effect on these parameter estimates. However, significant differences were found among the tillage practices with regard to  $C_0$  ( $P < 0.0001$ ), and between the N amendment treatments with respect to  $k$  ( $P < 0.0001$ ).

The average (across N amendment and soil depths) value of  $C_0$  for the NT runoff was 13.5 g CO<sub>2</sub>-C kg<sup>-1</sup> sediment and significantly higher than in the CT (6.9 g CO<sub>2</sub>-C kg<sup>-1</sup> sediment) and MP (7.7 g CO<sub>2</sub>-C kg<sup>-1</sup> sediment) runoffs. However, when expressed as a fraction of the total sediment C,  $C_0$  represents 32, 31 and 37%, respectively, of the total sediment C in the NT, CT and MP runoffs. A positive relationship ( $y = 0.27x + 1.18$ ,  $R^2 = 0.77$ , Fig. 2a) was found between sediment C( $x$ ) and  $C_0$ ( $y$ ). This narrow range ( $C_0$  as 31–37% of sediment C) and the linear relationship suggest that the relative proportion of eroded C susceptible to mineralization varies little among the three tillage practices considered. These findings

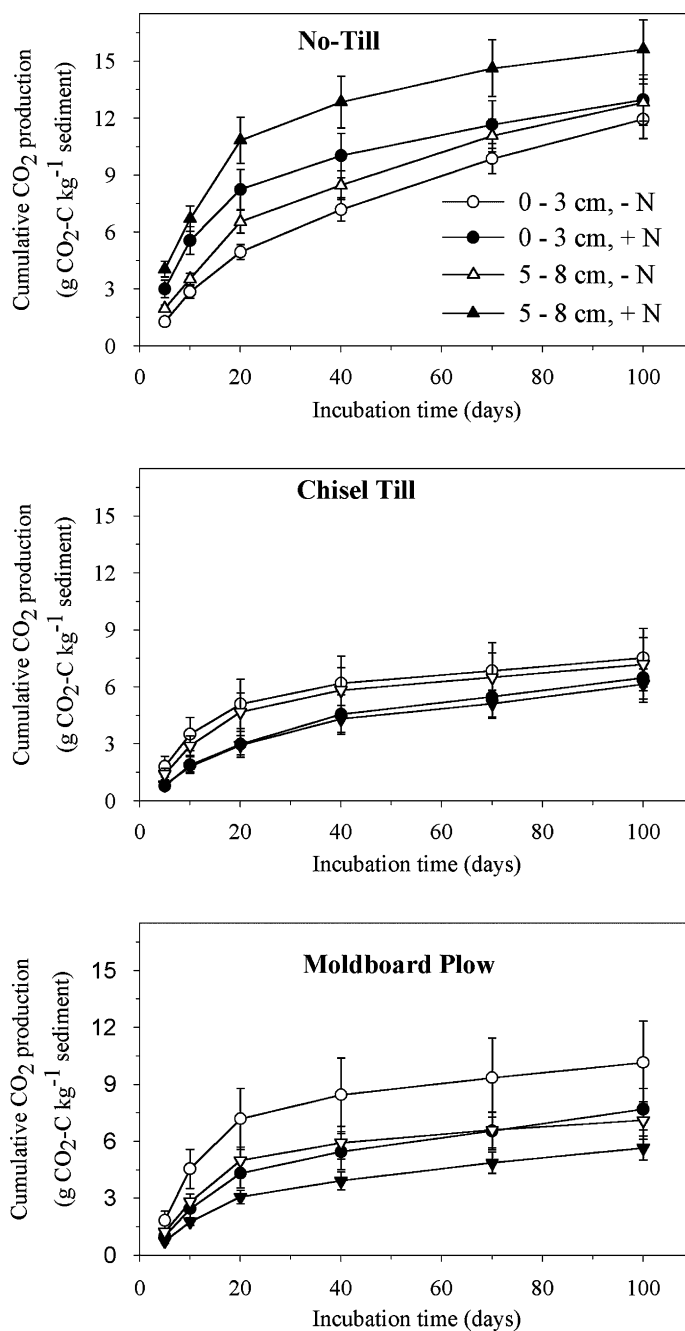


Fig. 1. Carbon dioxide (CO<sub>2</sub>) evolution in runoff from NT, CT and MP microplots. Runoff was generated through application of simulated rainfall (30 mm h<sup>-1</sup> for 1 h) to microplots constructed with the 0–3 and 5–8 cm layers of soil collected from long-term (38 years) tillage plots. Runoff sample was amended (+N, 1.25 g NH<sub>4</sub>-N ml<sup>-1</sup> runoff) or not (–N) and incubated for 100 days. The error bars represent the standard deviation of the mean ( $n = 9$ ).

Table 3

Estimates of potentially mineralizable C ( $C_0$ , g CO<sub>2</sub>-C kg<sup>-1</sup> sediment) and the mineralization rate constant ( $k$ , per day) in runoff through fitting of carbon dioxide evolution data to a first-order kinetic model

Tillage practices <sup>a</sup>	N addition	0–3 cm soil layer			5–8 cm soil layer		
		$C_0$	$k$	$R^2$	$C_0$	$k$	$R^2$
NT	–N	13.28 ± 0.80 <sup>b</sup>	0.021 ± 0.002	0.991	13.01 ± 0.68	0.030 ± 0.004	0.983
NT	+N	12.28 ± 0.47	0.054 ± 0.006	0.974	15.07 ± 0.39	0.059 ± 0.005	0.986
Tillage mean <sup>c</sup>		12.87 a	0.036 b		14.12 a	0.044 a	
CT	–N	6.61 ± 0.27	0.029 ± 0.003	0.990	6.17 ± 0.28	0.031 ± 0.003	0.986
CT	+N	7.16 ± 0.22	0.061 ± 0.006	0.980	6.94 ± 0.20	0.052 ± 0.005	0.985
Tillage mean		7.26 b	0.041 ab		6.62 b	0.040 a	
MP	–N	7.50 ± 0.40	0.037 ± 0.005	0.974	5.58 ± 0.29	0.035 ± 0.005	0.979
MP	+N	9.81 ± 0.37	0.058 ± 0.007	0.972	6.95 ± 0.25	0.053 ± 0.006	0.977
Tillage mean		8.70 b	0.044 a		6.29 b	0.043 a	

<sup>a</sup> NT: no-till; CT: chisel till; MP: moldboard plow.

<sup>b</sup> Values are mean ± S.D.

<sup>c</sup> Within a column, values followed by the same letter are not significantly different at the 95% level.

further support the concept of water erosion selectivity for labile soil organic compounds.

Irrespective of soil tillage and depth, amendment of runoff with mineral N stimulated runoff C mineralization, as evidenced by the higher rate constants (Table 3). Evolution of CO<sub>2</sub> in N-amended runoff samples proceeded at rates 1.5–2.1 times faster than in the unamended samples (Table 3). The effect of N amendment diminished after 20 days of incubation.

Cumulative runoff C mineralized during the 100-day incubation was significantly greater in NT (13.3 g CO<sub>2</sub>-C kg<sup>-1</sup> sediment) than in CT and MP runoffs. Cumulative runoff C mineralized in CT (6.8 g CO<sub>2</sub>-C kg<sup>-1</sup> sediment) and MP (7.5 g CO<sub>2</sub>-C kg<sup>-1</sup> sediment) was not statistically different (Table 4). Between 39 and 60% of the runoff C which underwent mineralization during the entire 100-day incubation occurred during the first 20 days (Fig. 2b). This observation is consistent with computed (using data in Table 3) runoff C

Table 4

Cumulative runoff C mineralized as affected by tillage practice and mineral nitrogen amendment

Tillage practices <sup>a</sup>	N addition	Cumulative runoff C mineralized (g CO <sub>2</sub> -C kg <sup>-1</sup> sediment)		Eroded C mineralized (%)
		0–20 days	0–100 days	
NT	–N	5.8 ± 1.7 <sup>b</sup>	12.4 ± 2.8	29
NT	+N	9.5 ± 3.5	14.3 ± 4.3	33
Tillage mean <sup>c</sup>		7.6 ± 3.3 a	13.3 ± 3.8 a	
CT	–N	3.0 ± 1.7	6.3 ± 3.0	28
CT	+N	4.9 ± 3.3	7.4 ± 4.2	33
Tillage mean		3.9 ± 2.8 b	6.8 ± 3.7 b	
MP	–N	3.7 ± 1.8	6.7 ± 2.8	35
MP	+N	6.1 ± 3.7	8.6 ± 5.0	46
Tillage mean		4.9 ± 3.1 b	7.7 ± 4.2 b	

<sup>a</sup> NT: no-till; CT: chisel till; MP: moldboard plow.

<sup>b</sup> Values are mean ± S.D.

<sup>c</sup> Within a column, values followed by the same letter are not significantly different at the 95% level.

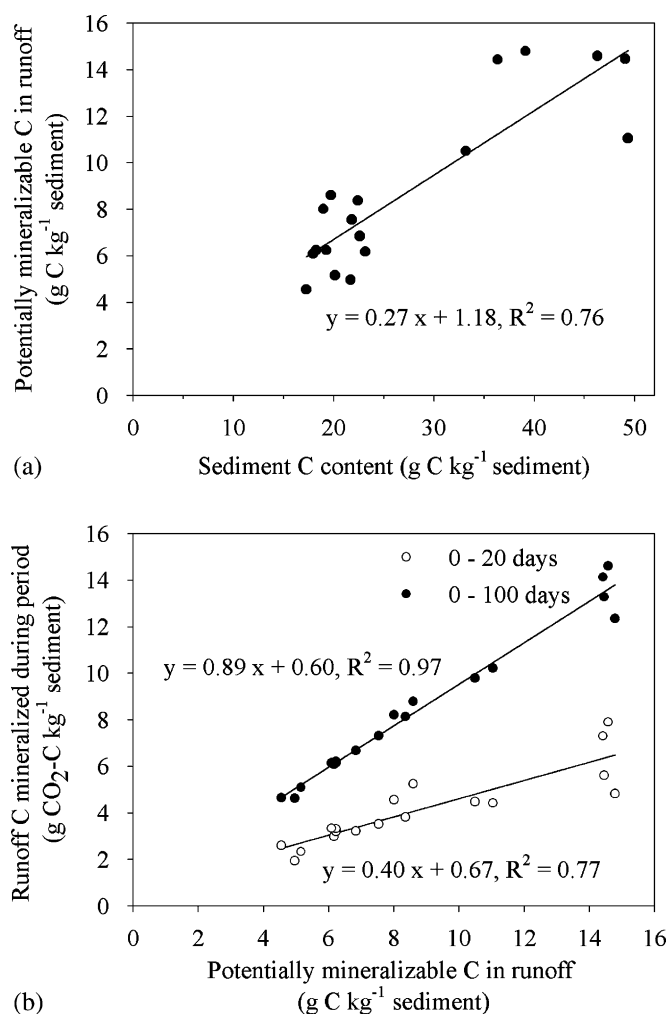


Fig. 2. Relationships between potentially mineralizable carbon ( $C_0$ ) and sediment C content (a), and cumulative runoff C mineralized in 20 and 100 days of incubation (b).

half-lives ( $t_{1/2}$ ) which ranged between 12 and 33 days. Strong relationships ( $R^2 = 0.77$ – $0.97$ ) were found between potentially mineralizable C in runoff and cumulative C mineralized during the 0–20 days and the entire 100-day incubation period.

## 4. Discussion

### 4.1. Extrapolating simulated C export to field scale

In this investigation, microrunoff plots (0.12 m<sup>2</sup>) and laboratory incubation were used to examine the

transport and fate of SOC mobilized during rainfall events. Concerns and questions which typically arise with this type of approach center around its representativeness of field conditions and the extent to which the data obtained with these microcosms can be extrapolated. To be able to compare this study results with field data, measured sediment delivery rates were expressed in g sediment mm<sup>-1</sup> rainfall. The data in Table 2 showed that the average C exports recorded during the present simulation study ranged between 0.02 and 0.04 g C m<sup>-2</sup> mm<sup>-1</sup> rainfall.

Lowrance and Williams (1988) applied simulated rainfall at an intensity of 63.5 mm h<sup>-1</sup> to fallowed,



bare-bedded and peanut-cropped runoff plots ( $39.6 \text{ m}^2$ ) and reported average erosion SOC exports in 30 min rainfall events corresponding to 0.08, 0.15, and  $0.07 \text{ g C m}^{-2} \text{ mm}^{-1}$  rainfall, respectively. Although the total amount of rainfall applied in the two studies were similar ( $\sim 30 \text{ mm}$ ), the higher export rates reported by Lowrance and Williams (1988) probably reflect the higher rainfall intensity. Using the sediment delivery rates observed in the study (expressed as  $\text{g sediment mm}^{-1} \text{ rainfall}$ ) and taking the long-term average annual precipitation ( $1060 \text{ mm}$ ) at the South Charleston Research Station, erosion rates of 5.7, 16.3 and  $11.4 \text{ Mg soil ha}^{-1}$  per year can be arrived at for the NT, CT and MP tillage practices, respectively. When the same computation technique is applied to the data reported by Choudhary et al. (1997) from a rainfall simulation experiment (rainfall event: 50 min at  $50 \text{ mm h}^{-1}$ ) conducted with Wooster silt-loam soils under long-term (32 years) tillage management, estimates of runoff soil loss obtained are also within similar ranges (NT:  $2.2 \text{ Mg soil ha}^{-1}$  per year, CT:  $6.3 \text{ Mg soil ha}^{-1}$  per year, MP:  $9.6 \text{ Mg soil ha}^{-1}$  per year, with annual rainfall =  $905 \text{ mm}$ ).

Estimates of erosion soil losses obtained in these simulation studies are probably slightly higher than under field conditions due to (i) positioning, during the simulation study, of the runoff trays at slopes (9% in this study) that are higher than in the field, (ii) compaction and hydraulic interruption caused by the soil block extraction procedure resulting in reduced infiltration and increased runoff, and (iii) the absence of a vegetation canopy and the removal of surface residue cover which absorbs the kinetic energy of raindrops (Parsons et al., 1994), reduces flow velocity (Brown and Norton, 1994) and provides an additional protection against erosion. Nonetheless, these estimates are reasonable and do not deviate greatly from erosional soil losses recorded ( $<0.1$ – $7.4 \text{ Mg soil ha}^{-1}$  per year, Edwards and Owens, 1991; Hao et al., 2001) in several areas across the state of Ohio. On the other hand, it is also important to recognize that, in the field, in addition to released SOC, runoff water also contains C originating from plant canopy (throughfall and stemflow) and decomposing litter. As these C sources tend to be highly labile (Qualls and Haines, 1992; Yano et al., 2000), they could therefore augment the proportion of mineralizable C in runoff, although the total C export may

be reduced. Whether this is significant at the field level is unknown at the present time.

#### 4.2. Labile C fractions in runoff as affected by soil management

Erosion control is one of the many benefits of no-tillage widely documented in the literature (see review by Cannell and Hawes (1994)) and confirmed by the data obtained in the present research. Sediment production in the NT soil was 2–2.8 times lower than in the other tillage practices and was likely controlled by soil structural stability as suggested by the inverse relationships between sediment output and soil aggregation indices (Tables 1 and 2). However, even though lower erosion rates were recorded in the NT soils, they had the highest cumulative  $\text{CO}_2$  production ( $13.3$  versus  $7.3 \text{ g CO}_2\text{-C kg}^{-1}$  sediment in the other soils) due to the combined effects of higher C content of sediment ( $42.3$  versus  $20.3 \text{ g C kg}^{-1}$  sediment) and the labile character of the C fractions removed (average  $C_0$  was 1.8 times higher in sediment from NT than that from the other tillage treatment). The labile character of runoff C is an important finding which confirms the high selectivity of erosion for the labile SOC fractions. It also suggests that, although management practices such as no-tillage results in SOC accumulation and significant reduction in soil erosion, even small erosional C leakage from these systems could have measurable consequence on atmospheric  $\text{CO}_2$  as the C mobilized is very prone to mineralization.

Mineralization of runoff C increased with mineral N amendment suggesting that fertilization timing, rainfall distribution and seasonal variation in soil erodibility could have some impact on the fate of eroded C at the field-scale level. Mineral N application rate in the study (equivalent to  $0.15$ – $0.20 \text{ g N kg}^{-1}$  soil), although high, is not uncommon in recently fertilized soils (Gascho et al., 1998) and at spring thaw (Roth and Fox, 1990; Kessavalou and Walters, 1999). The high initial rate ( $<20$  days) of runoff C mineralization suggests that under field conditions, most of the erosion-induced  $\text{CO}_2$  production will occur contemporaneously and in the days immediately following rainfall events, and therefore strategies for monitoring erosion-related  $\text{CO}_2$  fluxes must be tailored accordingly. This requirement becomes more important when mineral N availability is greater.

Climatic variations and geomorphological features are factors which could influence the fate of eroded C at the field and watershed scales, but these impacts could not be evaluated due to limitations of the runoff simulation setup used in the present investigation. Water erosion is a multi-process phenomenon involving the detachment of soil aggregates, and the transport of suspended soil particles and associated C. Eroded materials are either delivered to water bodies with runoff, retained in terrestrial deposits, or be the subject of a series of deposition–resuspension events (Hairsine et al., 1999) over the eroding landscape depending on rainfall characteristics and landscape patterns. These deposition–suspension events combined with temperature variations, are likely to affect erosion-induced CO<sub>2</sub> production in the field in ways that could not be reproduced in the simulation study. These limitations therefore call for additional field-scale investigations of exports and mineralization of eroded C so that assessments of water erosion impacts on global C cycle can be improved.

## 5. Conclusions

The transport of soil organic carbon by water erosion and its mineralization was investigated. Soil tillage was an important factor controlling sediment production, and export of sediment-associated C. Consistent with the first hypothesis of the study, potentially mineralizable C and cumulative C mineralized during the incubation was greater in runoff from NT than from the other tillage systems. However, due to selective removal of biologically active C fractions by water erosion, the labile C fraction in sediment C was relatively constant (31–37%) among the tillage practices and at the soil depth (0–8 cm) considered. Kinetically derived estimates of potentially mineralizable C in runoff were highly correlated with cumulative runoff C mineralized which amounted to 29–35% of the total sediment C in the absence of mineral N amendment. With N amendment, the amount of runoff C mineralized increased to 33–46% of the total sediment C, consistent with the third hypothesis of the study. Although field-scale research is needed to refine this assessment, the present study demonstrates the release of labile C by water erosion, and the potential for additional CO<sub>2</sub> input into the atmosphere resulting

from mineralization of the C mobilized during erosional disturbances.

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